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Overview Article

Functional assessment of a reference wetland set as a tool for science, management and restoration

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Abstract. Wetlands are increasingly becoming the target of efforts to restore or mitigate past and current loss of area and other impacts on their function. Tidal wetlands serve an array of functions deemed beneficial (ecosystem services) but there are relatively few efforts to provide verified indicators of these functions or assess variability in function among wetlands. We assessed twelve functions ranging from wave energy dissipation to fish species richness in tidal freshwater wetlands on the Hudson River. These functions were assessed along with potential "indicators" of function at fifteen marshes selected to span hydrogeomorphic classes as well as expected level of function. Functions varied dramatically among wetland sites, with scores summed across functions ranging from 16% to 70% of the maximum possible. Some of the functions were positively associated such that improvement in one would probably be accompanied by improvements in others. Some functions (e.g., surface water exchange and breeding bird habitat) were negatively correlated indicating that one site cannot maximize all potential functions. A verified reference data set allows more objective selection of targets and sites for restoration as well as establishing realistic goals for what might be achieved. The validated indicators of function are valuable tools for extrapolating from a few intensively studied sites to the larger, unsampled, population of wetland sites in a region.

Key words. Wetlands; functions; hydrogeomorphic; tidal freshwater marshes.

Introduction

Wetlands are recognized as a particularly important and valuable element of most landscapes, performing a host of ecological functions and providing "ecosystem services" (Costanza, et al. 1997; Keddy, 2000). Upland, riverine, palustrine and tidal wetlands often support high rates of primary production (Brinson et al., 1981), are significant habitats for various species of wildlife (Odum et al., 1979), mediate a range of biogeochemical transformations leading to improved water quality (Johnston et al., 1990; Jansson et al., 1998) and have high value to many members of society. Despite their perceived value (occasionally quantified, see Stevens et al., 1995), wetlands in the United States and around the world are still being impinged upon by changes in landuse/landcover in their catchments and sometimes drained or filled (Mitsch and Gosselink, 1993; Moser et al., 1996). Aside from this continuing degradation and loss of wetlands, our ability to protect, manage and restore these systems remains fairly poor, largely because we do not have tools for rapidly yet plausibly assessing their value. Wetlands that might be particularly important in a region are not necessarily recognized since many of the functions (e.g., habitat for rare plants, microbial removal of nutrients from

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surface waters) do not lend themselves to rapid quantification in the field. At the other extreme, wetlands functioning relatively poorly or even wetlands performing functions at levels common in the region would represent less of a net loss if unavoidable impacts could be directed to those sites.

In large part as a response to the continued loss of wetland areal extent and function, efforts to restore degraded wetlands, mitigate for damage and even create new wetlands is a huge business in North America and elsewhere (see NRC, 1992; NRC, 2001; Mitsch et al., 1998; http://www.epa.gov/owow/wetlands/restore/). Efforts to restore or mitigate are often unsuccessful (Zedler, 1999) at least in part because of unrealistic (or even unstated) goals for what wetlands in a region are capable of producing or supporting (Brinson and Rheinhardt, 1996). Meeting scientifically designed goals may be expensive (Kiviat et al., 2000). Moreover, monitoring of restored/mitigated/created wetlands is often short-lived due to the expense of properly assessing the multitude of functions carried out by natural wetland ecosystems (but see Galatowitsch and van der Valk, 1996; Wilson and Mitsch, 1996).

Aside from management needs, the science of ecology is struggling with appropriate approaches to extrapolate what is known about a few well-studied sites to the larger regional context (Groffman and Likens, 1994; Cole and Brooks, 2000). The average study area is generally less than 100 m in diameter (e.g., Kareiva and Andersen, 1986) and study duration is typically less than 2 years (Tilman, 1988). The small-scale and short duration studies mean that the vast majority of wetland areas have never actually been sampled yet we are asked to develop an understanding of ecosystem function applicable to the entire population of wetlands or other ecosystem types. There is often an understandable hesitancy in applying conceptual or empirical models developed for certain sites to the rest of the population.

For all the above reasons, simple yet verifiable approaches for quantifying wetland performance have been developed with the intent of allowing a broader-based approach, spatially and temporally, to the management and scientific study of wetland ecosystems (Gwin et al., 1999; Hruby et al., 1995). The proliferation of these approaches in the United States has largely come from the regulatory mandate to assess individual sites prior to making decisions on permit applications. Additionally, there is a perceived need for survey-level inventories of what wetlands are present in an area, how their functions vary and which factors may be driving the observed variability in function. These approaches range from assessment of habitat suitability for single species (often those of conservation concern or particular economic value) to a broad, arguably more objective, effort to describe all the important functions.

In this paper we describe the steps involved in carrying out a relatively broad functional assessment of tidal freshwater wetlands in the Hudson River, New York, USA. In general ecological usage the term function refers to a process or transformation of materials or energy although the term has become blurred somewhat in practical usage to include habitat characteristics (e.g., the function of providing habitat) and sometimes values (e.g., the function of providing recreation). Many functional assessments combine social values with ecological functions or only consider functions with a clear social value (NRC, 2001). While a functional assessment should probably focus on ecological functions or their very close surrogates (i.e., peak standing crop as surrogate for primary production) it is important to recognize the social context and values associated with decisions regarding wetland management.

Our goals are to: (1) highlight important decisions that must occur fairly early in the process and that affect the utility of the final product, (2) present the information we collected to characterize the various functions and in some cases validate simpler indicators of those functions and (3) show how the information can be used to address scientific and management issues. Our intent is to highlight critical issues, not to present all the details and documentation necessary to plan or carry out such an assessment.

Approach and methods

Selection of sites and functions

There are many wetland assessment procedures, some targeted at individual species and some encompassing functions at multiple scales including physical, chemical and biological processes. Most of the common approaches were recently compiled by Bartoldus (2000) who provides guidelines for selecting a procedure depending on specific needs. A national guidebook for assessing river floodplain wetlands is available (Brinson et al., 1995) and served as a point of departure for our design of tidal freshwater wetland assessment. We adopted and modified the Hydrogeomorphic (HGM) Functional Assessment (see Brinson and Rheinhardt, 1996) because it specifically recognizes differences in the types of functions attributed to different classes of wetlands, i.e., tidal wetlands provide different functions from bogs (mires). Most importantly, it forces an explicit statement of what the important functions are for the wetlands under consideration. This approach removes some (but not all) of the subjectivity in wetland assessment and in our application gives each function equal weight in the final accounting. Some very specific issues such as habitat for a legally protected endangered species or remediation of contaminated sediments should be dealt with separately rather than included with the more coarse-scale functions typical of a regional assessment. A more detailed description of the process is provided in Smith et al. (1995) or Brinson and Rheinhardt (1996).

Two important decisions are made early in the process of assessing wetland function and it is common to assemble a small working group representing different fields of expertise to make these decisions. First, the class(es) of wetlands to be assessed are described and this decision often narrows the list of potential functions that will need to be considered. For instance, bogs do not provide a significant floodwater protection function while this is an important "service" provided by riverine wetlands. In our case we concentrated on freshwater tidal marshes of the Hudson River, excluding for example, brackish marshes (10 ppt or higher maximum salinities) and tidal wooded swamps. This decision was based on the areal predominance of this class of wetland, availability of information and management interest in conserving and restoring these marshes.

Once the class of wetlands is selected, the working group has to decide on a list of the most important functions. The list should cover a range of functions, including biotic, physical, possibly social and economic. Given logistic and financial constraints and the need to focus on the most important functions the list will be reasonably short with, for example, a recent test of consistency among practitioners considering 14 functions (Whigham et al., 1999). It is important that the decision process be clearly documented since some functions will have to be neglected and these decisions must not be perceived as capricious. The working group should include a diversity of expertise so as to be able to identify and describe in rigorous terms functions spanning the range from, for instance, plant productivity to floodwater retention. Ideally the list of functions together with the rationale for their inclusion will receive broad review by managers, scientists and interested groups prior to actual data collection. The working group and others must realize that the selection of functions circumscribes the vision of what a "valuable" wetland will include and so these decisions must be as clear and well documented as possible. In general, all the functions to be included are given equal weight so that, for example, vertebrate habitat is no more or less important than the ability of a marsh to provide detritus. Obviously the working group can decide to apply a weighting factor to certain functions and again, this is a subjective decision requiring a clear rationale and documentation to help avoid conflict after data are collected. We began the selection of functions using the list in Brinson et al. (1995) but we separated some functions. For example, we considered nutrient retention in plant biomass and nitrogen removal via denitrification separately rather than the more inclusive "Characteristic Nutrient and Elemental Cycling" of Brinson et al. (1995).

Site selection is extremely important since the goal is to have the sampled set of wetlands be representative of the entire population of wetlands in a given class but logistics (distance, physical access and permission) often represent a serious constraint. Ideally one would randomly select a large enough number of sites to span the expected range in function. One of the utilities of the reference data set (described below) is to characterize the "best" and "worst" wetlands in the region so the sites should span the entire range of function. Because the distribution of function is unlikely to be uniform (more likely to be strongly skewed) a random sample would

Table 1. Location and general description of reference marshes on the Hudson River in New York State. River km = distance in river kilometers from The Battery, New York City; Salinity = approximate maximum salinity; RR (km) = perimeter of marsh in km bordered by railroad; Exposure = % of marsh perimeter directly exposed to river.

Reference Marsh	Abbreviation	Type	USGS[a] Quadrangle	County	River km	Salinity (ppt)	RR (km)	Area (ha)	Exposure %
Hell Gate	HG	sheltered	Ravena	Rensselaer	216	0	0.0	13.1	11
Schodack Island	SI	fringe	Ravena	Rensselaer	213	0	0.0	1.1	43
Mill Creek	ML	fringe	Ravena	Columbia	206	0	0.0	0.2	50
Little Nutten Hook	NH	sheltered	Hudson N	Columbia	198	0	0.0	4.0	36
Stockport Flats	SF	sheltered	Hudson N	Columbia	195	0	1.9	49.3	11
Unnamed Island	UI	fringe	Hudson N	Columbia	195	0	0.0	0.9	50
Rogers Island	RI	fringe	Hudson S	Columbia	185	0	0.5	47.8	49
Cruger Island North	CI	sheltered	Saugerties	Dutchess	159	0	0.3	2.7	5
Cruger Island South	CS	sheltered	Saugerties	Dutchess	159	0	0.2	6.3	2
Tivoli North Bay	TN	enclosed	Saugerties	Dutchess	159	0	2.2	110.2	1
Vanderburgh Cove	VB	enclosed	Kingston	Dutchess	142	0	1.5	18.7	1
Cornwall Bay	CB	fringe	Cornwall	Orange	92	1	0.0	0.7	56
Moodna Marsh	MD	enclosed	Cornwall	Orange	92	1	0.4	16.2	4
Constitution Marsh	CM	enclosed	West Point	Putnam	85	3	2.0	67.3	1
Manitou Marsh	MN	enclosed	Peekskill	Putnam	77	5	1.8	22.4	1

[a] USGS: United States Geological Survey.

probably not include the extremes. The original proponents of this approach suggest selection of what they call "reference standards" which are the best sites for a particular function or groups of functions (see Rheinhardt et al., 1997). Hudson River marshes have been intensely affected by human activities for a long time so we were not comfortable designating reference standards prior to data collection. We made an effort to select sites spanning the range of apparent human impact recognizing that this is a subjective process.

We had three hydrogeomorphic subclasses (Enclosed, Sheltered and Fringe marshes) describing degree of isolation from the mainstem river and presumed hydrodynamic energy. Fringe marshes are unprotected and often occur on sandy dredge spoil while all our enclosed marshes occurred behind the railroad embankment. Hydrology is likely to vary significantly among classes of wetlands (Cole et al., 1997) often leading to the expectation of different levels of function. There were five marshes within each subclass and these were selected to span the range of anticipated function resulting in fifteen marshes split among three subclasses (Table 1). All functions could not be assessed at all sites, for example there are 5 missing values for BREEDING MARSH BIRD HABITAT.

Field sampling

Sampling of individual marshes was organized along triplicate transects spanning mean low water (MLW) to mean high water (MHW) and for consistency all transects were established by a single individual. Five sampling plots (0.25 m^2) were arrayed from low to high elevation along the transect so as to represent the major sub-habitats (e.g., shallow subtidal to emergent marsh). Certain variables were only measured at particular elevations (e.g., submersed vegetation only at lowest elevation). At each plot basic data on plant species composition, aboveground biomass, plant nitrogen and phosphorous content (Templer et al., 1998), stem density and length were collected. Additionally, coarse woody debris (diameter > 1 cm) and litter cover were estimated visually and dry mass of litter measured for each plot. Soil samples (3.5 cm diameter, depth of 15 cm) from each plot were analyzed for color (Munsell Soil Color Charts), sand content (dry mass of particles < 2 mm and > 0.064 mm) and organic content (loss on ignition at 450 °C for 4 hr). Denitrification potentials (Smith and Tiedje, 1979) were assayed at emergent marsh plots at 12 of the 15 sites. Benthic invertebrates in low intertidal and subtidal sediments (five replicate cores per plot, 3.5 cm diameter cores to 15 cm depth, 0.5 mm sieve) and the amphipod (*Gammarus* spp.) associated with submersed vegetation (rinsed from vegetation three times, collected on 0.5 mm sieve then animals and plants dried and weighed) were collected for each transect. Fishes were collected with nets (7.4 mm nylon mesh) staked across rivulets draining the marsh surface and seines (6.1 m long, 1.2 m deep, 7.4 mm nylon mesh) in low intertidal and subtidal areas. There were several variables obtained from topographic maps (United States Geological Survey, 1 : 24,000 Quadrangles) or aerial photographs including marsh area, perimeter, degree of exposure to the main channel of the river and area of different vegetation zones.

Data analysis

We had indirect measures (potential indicators) for all twelve functions (Table 2). For seven of the twelve we also had direct, independent measures of the function. For example, we could not directly measure surface water exchange for each site and used three indirect measures (exposure to the mainstem river, presence of physical barriers and truncation of tidal range) to estimate surface water exchange. For the seven functions where we have direct measures of function and independent indirect measures we constructed statistical models to ascertain how well our indirect measures predicted the direct measures. All three subclasses were combined for all analyses to provide a sufficient number of observations for reasonably powerful statistical analyses.

The triplicate transects within a site did not differ significantly (ANOVA, $p > 0.05$) and so variables were averaged across transects before calculating scores for the functions. For purposes of estimating a total score for each marsh and to allocate equal weight to the various functions, all the functions were scaled to vary between 0 and 1. In general sites scoring higher are considered to be functioning better and all our functions were defined such that scores of 1 represent "best" conditions. In other applications it is possible that large positive values (e.g., rapid sediment accumulation due to erosion of uplands) may represent poor functioning or disturbed conditions so one could simply switch the sign after scaling and these would have a negative effect on the total score for a site. Additionally, if an intermediate quantity was perceived to have greatest value (i.e., 50% plant cover for habitat) one could calculate the absolute value of the difference from 50% so that either more or less cover would lead to lower scores, i.e., SCORE = 1 – ((|50-observed|)/50). For directly-measured functions we scaled results for a particular site by subtracting the minimum observed for that function from the value at that site and dividing by the range. Thus, the scaled value $V_s = (V_{observed} – Min)/(Max – Min)$ and will range from 0 to 1. For indirectly-measured functions we scaled the input variables (i.e., exposure, barriers and truncation in the case of surface water exchange) to allow indirect measures with differing numerical ranges to contribute equally to variation in the function.

Table 2. List of functions with brief description. Indirectly-measured functions are estimated from surrogate variable judged to be related to the actual function. These models are shown following the function definition. Directly measured functions do not require surrogates or indicators but these are useful for surveying these functions at large numbers of sites.

INDIRECT

Function	Description	Indirect Measure
ENERGY DISSIPATION	Ability of the marsh to lessen the erosive impact of water and wind energy from the river on the upland shoreline or other marshes.	$ED = \frac{(SAV + VL + VU + WL + WU)}{5} \times E$ where SAV = volume of submersed vegetation, V = volume of intertidal vegetation and W = wood debris cover in the Lower and Upper intertidal zone. E = proportion of marsh perimeter open to tidal influence.
SURFACE WATER EXCHANGE	The circulation of water into, through, and out of a marsh.	$SWE = \frac{(E + B + T)}{3}$ where E = exposure, B = 1 – proportion of marsh affected by barriers and T = 1 – proportional truncation of tidal range.
MUSKRAT HABITAT	Availability of high quality food plants, and soil texture suitable for maintaining burrows.	$MH = \frac{(PV - S)}{2}$ where PV = preferred vegetation (area of *Typha* plus *Acorus*) and S = percent sand.
BREEDING MARSH BIRD HABITAT	Percent of the marsh dominated by types of vegetation preferred by birds that are specialized to breed in marshes.	$BMB = \frac{(CA + (PLM \text{ or } SW \text{ or } PH))}{2}$ where CA = cover of cattail and term in parentheses is cover of one or more of *Lythrum salicaria* mix, sweetflag, or *Phragmites australis*.
FOOD BASE FOR DUCKS AND RAILS	Abundance and accessibility of seeds and vegetative parts of selected food plant species.	$FDR = \frac{\left[PF1 + PF2 + \left(\frac{PF3}{2}\right)\right]}{3}$ where PF1 = stem density of emergent plant species that are preferred food, PF2 = biomass of submersed plant species that are preferred food, and PF3 = less-preferred species of submersed plants.

DIRECT

Function	Description
PLANT BIOMASS	Aboveground peak standing crop.
NUTRIENT RETENTION	The ability of a marsh to remove nutrients from surface waters via assimilation into plant biomass.
NITROGEN REMOVAL	Potential denitrification.
FISH RICHNESS – LOW	Number of non-migratory fish species that utilize the tidal creeks and Lower intertidal zone for feeding and shelter.
FISH RICHNESS – HIGH	Number of non-migratory fish species that utilize the Upper intertidal zone for feeding and shelter.
FUNDULUS	Density of *Fundulus* that utilize the Middle and Upper intertidal zone for feeding and shelter during high tide.
INVERT RICH	Number of higher taxa of macroinvertebrates within the Lower intertidal and subtidal marsh habitats.

Results

A premise of the HGM functional assessment is that sites actually sampled will encompass the full range of functions within the region. Almost all our direct and indirect measures of function varied broadly among sites (Fig. 1) and the quartile range (75^{th} – 25^{th} percentile) was greater than the median for nine out of twelve functions. As expected, there were statistically significant differences among our HGM subclasses (fringe, sheltered, enclosed) for ten out of twelve functions (Fig. 2). In nine of these ten cases, the fringe marshes were different from the sheltered and enclosed while these two subclasses did not differ from each other so the contrast between fringing marshes and protected/enclosed marshes generates the broadest range in the data set. Only for nitrogen removal

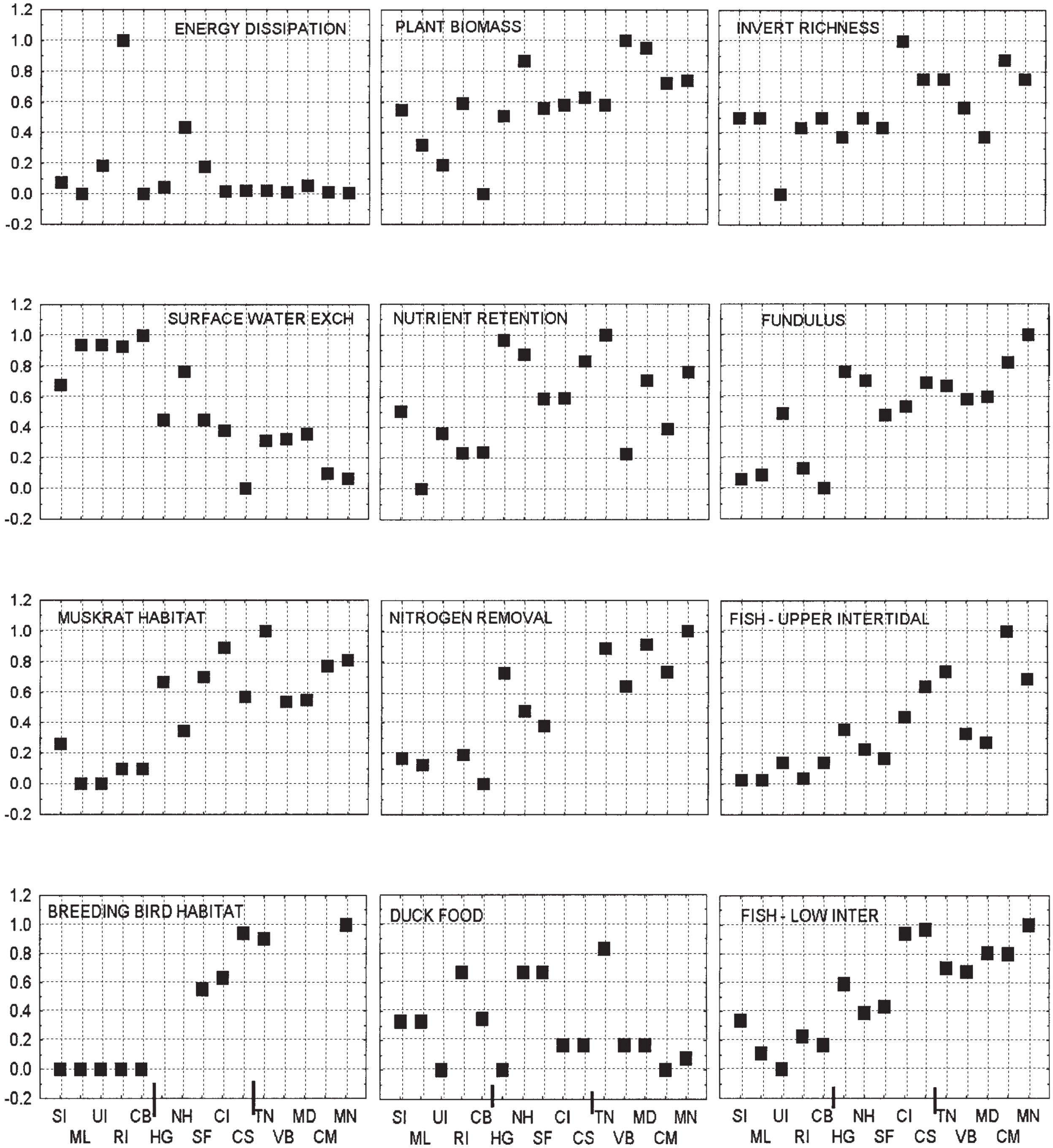

Figure 1. Values for functions at fifteen tidal wetland sites on the Hudson River. Functions and abbreviations are described in Tables 2 and 1, respectively. Sites are grouped by sub-class, left-most are fringe, middle are sheltered, right-most five are enclosed.

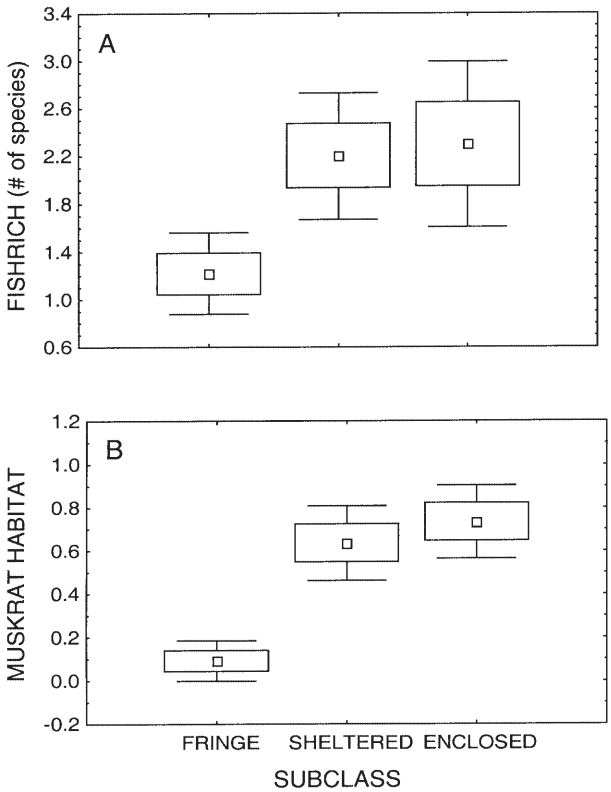

Figure 2. Variation in fish species richness (A) and muskrat habitat (B) among subclasses of wetlands. Values are (A) mean number of fish taxa (± 1 and 2 SE, box and whiskers respectively) for a marsh site or (B) scaled indirect measure of muskrat habitat suitability. For these examples functions differed significantly ($p < 0.05$) with Fringe marshes differing from Sheltered = Enclosed.

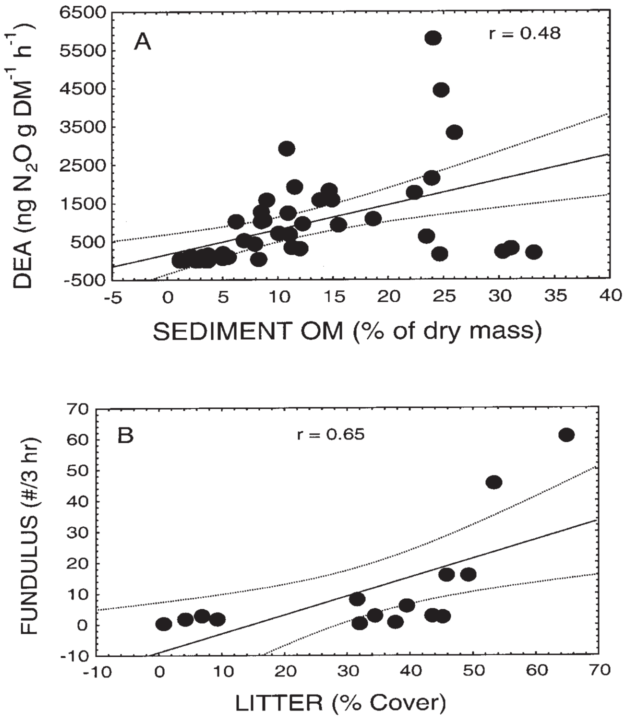

Figure 3. Relationship between indirect measures (indicators) of function and separate direct measures for (A) nitrogen removal (potential denitrification, DEA – denitrification enzyme activity) and sediment organic matter (% of dry mass) and (B) *Fundulus* abundance (catch per 3 hr deployment of block net) and % litter cover. For (A) each point represents a separate sediment sample. For (B) the sampling locations for litter and *Fundulus* do not coincide so the relationship is derived from site averages.

Table 3. Directly measured functions along with indirect measures (indicators). For these seven functions there were statistically significant correlations with at least one indirect measure. Values are Pearson Product Moment correlation coefficient and p value. N/A = not applicable since the indicator (plant biomass) is itself predicted from surrogates (plant length).

FUNCTION	INDIRECT MEASURE	Coefficient	p
PLANT BIOMASS	PLANT LENGTH	0.93	< 0.001
NUTRIENT RETENTION	PLANTBIOMASS	N/A	N/A
NITROGEN REMOVAL	SED WATER	0.61	< 0.001
	SED OM	0.48	< 0.001
FISH – LOW	SED OM	0.73	0.002
FISH – HIGH	SED OM/SAND	0.64	0.005
FUNDULUS	LITTER	0.65	0.008
INVERT RICH	SED OM/SAND	0.81	0.0002

(denitrification potential) were sheltered marshes significantly different from enclosed marshes.

For all of the seven directly measured functions we confirmed independent measures (simpler indicators) for those functions (Table 3, Fig. 3). For example, denitrification is probably an important process of nitrate removal in most wetlands yet even the simpler potential assays we conducted are fairly labor intensive and demand specialized instrumentation. Sediment organic matter or water content accounted for 23 or 37% respectively of the variation in denitrification potential in our data set making these simpler measures useful as predictors in a broad survey of denitrification potential among sites within our reference domain.

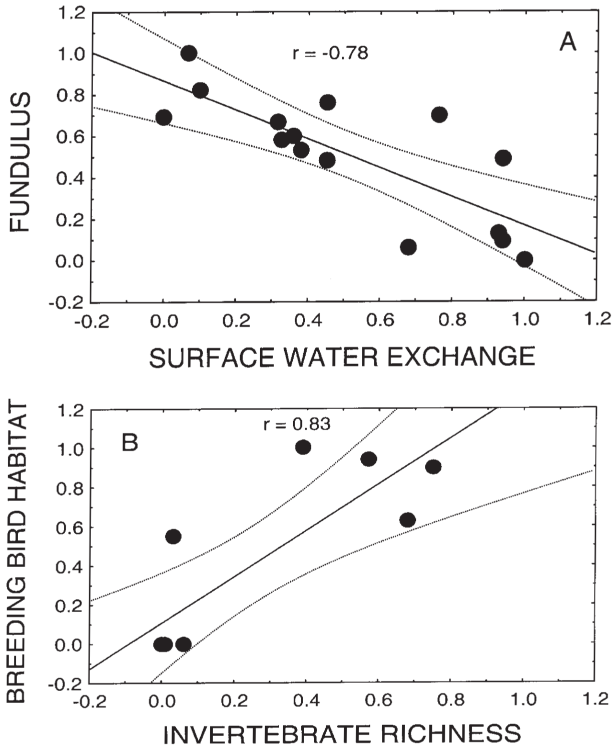

Figure 4. Examples of relationships among functions showing both negative (A) and positive (B) associations among functions. All functions have been scaled (0 – 1), n = 10 for BREEDING BIRDS due to missing values, some points overlap.

There were significant correlations among functions with eleven of twelve functions being correlated with at least one other function (e.g., Fig. 4). Most of the significant correlations were positive, i.e., an increase in a function would be accompanied by an increase in at least one other function. The negative correlations all involved SURFACE WATER EXCHANGE, which characterizes the exposure regime and differs dramatically among HGM sub-classes. The high scores for SURFACE WATER EXCHANGE in fringe marshes are associated with low values for plant biomass and sediment organic matter, which are two strong driving variables in many of the other functions. A Principal Components Analysis also revealed clustering of functions, specifically SURFACE WATER EXCHANGE loaded negatively on the first principal component while many of the organismal functions (fishes, invertebrates, muskrats) loaded positively.

Collection of our data set required four field and laboratory personnel for a two-month period in each of two years, and approximately one person-year of sample analysis and data analysis in addition to time for discussions and decision-making among the team of experts. Realistically the effort is likely to take two full years from initiation to completion with a total cost of 100,000 US$.

Discussion

A set of reference data on wetland functions should contain a reasonable span of values and describe apparent differences among wetlands and can be useful in setting restoration targets and success criteria (Rheinhardt et al. 1997). Our data set varied broadly among sites and in general documented the perceived differences among wetland subclasses. As such it should provide a useful context for site-specific studies of specific processes and assist with individual management decisions. At a minimum it provides a relative measure of how a particular marsh compares to others in the reference domain. For instance a measure of fish species richness at an individual site under study or a site awaiting a management decision could be cast as high, medium or low for the region. Also, if some management action or experiment designed to examine controls on fish species leads to a given change in richness one could state whether such a change was large or small relative to regional variability in fish species richness. Similarly, there is considerable interest in temporal variability in wetland function (Morris and Haskin, 1990; Newell, 2001) and the reference data set provides a reasonable coverage of local spatial variability for purposes of comparing variation in time and space. For all these reasons simply having a broad assessment of function across diverse sites is a valuable resource.

Beyond simply measuring functions the approach encourages the practitioner to propose and validate simpler indicators of these functions. These indicators (when validated) allow extrapolation to a much larger number of sites greatly expanding the power and resolution of the reference data set. Moreover, these indicators are often potential controlling factors of the functions of interest and documentation of significant correlations suggests important regulators of functions. For our seven directly measured functions we validated simpler indicators that were capable of accounting for ~ 40 to 90 percent of the variation in the function. In some cases, e.g., denitrification potential there may be a fairly direct cause and effect link between the indicator (soil organic matter or water content) and the actual function (c.f. Groffman, 1994) but in general it is best to view the indicators as associations rather than causative. Sediment characteristics appeared as useful predictors of several functions (Table 3) and sediment organic matter is typically an important target in wetland restoration efforts (Craft and Richardson, 1993; Bishel-Machung et al., 1996). Fish and macroinvertebrate taxa richness were both correlated with sediment organic matter but there is not necessarily a direct mechanistic linkage underlying the relationship. Lack of understanding of the mechanism does not reduce the utility of the indicator although understanding why a pattern emerges always engenders greater confidence in the generality of the result.

One application of the HGM approach to functional assessment is to provide relatively rapid description of particular sites under consideration for some management action, whether permission for some impact or targeting for protection or restoration. In most situations, such an application would use the indirect measures or indicators to assess a site and it must be recognized that the uncertainty for any of the relationships between indicator and direct measure was substantial. Coefficients of determination for the regressions ranged from ~0.3 to 0.8 and any given prediction could vary as much as two-fold (c.f. Fig. 3). The strength of this approach is the ability to describe a reasonably large number of sites with minimal effort recognizing the limitations for any site-specific observations. For decisions regarding large areas of potential impact or for monitoring of important sites, field experiments, etc. it is likely that the effort to directly measure functions instead of relying on indicators will be justified. The indicators may still be useful as "triggers" for example in monitoring a restoration site where the effort to directly measure functions would only be initiated when indicators approached some critical value. Similarly, indicators might be useful in selecting study sites where an investigator wants to sample across a gradient in some function with a reasonable *a priori* knowledge that the gradient exists.

The reference data set, even without models and indicators, also has specific utility in decisions about wetland protection and targets for restoration. With reference data, one can describe the "best" functioning and "worst" functioning sites in a region. For instance, the highest observed total score (summed across functions) was 70% of the possible maximum and the minimum was 16%. Sites at the upper end of the observed range may represent achievable targets for restoration goals or mitigation criteria in addition to being important sites for protection. Use of reference data would allow a manager to set the required level of achievement at a numerical value known to actually occur in the region under present conditions. While this avoids the issue of expecting (or demanding) unrealistic achievements for a restoration or mitigation project (such as striving to reattain pre-industrial conditions) it does admittedly constrain the ultimate target to something less than might be achieved under the perfect set of circumstances. At the other end of the spectrum, knowing which sites are functioning poorly suggests logical opportunities for restoration or enhancement since these sites have the greatest scope for improvement. Also, if the restoration effort causes damage to the site there is less functionality to be lost. The alternative view is that the poorest sites in a region may be under multiple stresses, some of which may be irreparable, and thus are poor candidates for potential restoration. Also, poor sites may be constrained by external factors (development in the watershed or alteration of water inputs) not amenable to restoration and such considerations would enter into the feasibility phase of restoration planning. In any case, knowing attainable values for a range of functions can only improve the confidence in site-specific decisions for management activities.

Aside from setting targets and selecting sites, the reference data set can help with resolving conflicts among functions and activities. Almost inevitably when managing large complex ecosystems with multiple uses there will be conflicts among groups with different objectives. For example, there were negative correlations (although non-significant) between the FOOD FOR DUCKS/RAILS function and all four of the functions related to fishes. Therefore, one might anticipate that user groups interested in one versus the other group of animals might lobby for enhancement of that function at the expense of the other. One way to approach resolution would be to assess the regional extent of duck versus fish habitat using the reference data. If highly functional duck habitat were more common in the region then managers might justifiably target fish habitat in restoration or protection programs.

Some management activities also have potentially conflicting outcomes. In the northeast United States the native yet invasive common reed (*Phragmites australis*) is frequently targeted for extermination (Marks et al., 1994). Cutting and herbiciding of reed leads to accumulation of ammonium in porewaters and a short-term decline in denitrification potential (Findlay et al., in press). Porewater ammonium may diffuse into the overlying water and a reduction in denitrification represents loss of an important sink for inorganic nitrogen so the net effect is that the site will be less of a nitrogen sink than sites with abundant reed. One goal of reed removal is to promote plant species richness (Farnsworth and Meyerson, 1999; Ailstock et al., 2001) so the trade-off is a reduction in the ability of the site to retain or process nitrogen in order to gain plant species richness. As for the example above, reference data on the prevalence of these two functions may indicate whether or not the trade-off is justified. If sites with high nitrogen retention are abundant relative to sites with high plant richness, then the trade-off seems justified. Conversely, if surface water nutrients are a serious problem in the region, it makes less sense to trade the ability to process nutrients in order to gain plant species.

While we see considerable value to carrying out a functional assessment on a set of reference wetlands there are several unresolved issues and opportunities for improvement. The first and perhaps most troubling is the subjectivity introduced in the choice of reference sites and functions. At this stage, larger sets of functions and clear statements of why the particular list of functions is deemed most important appear to be the best guard against overly narrow visions of what constitutes a desirable wetland. Perhaps as this approach is applied, relatively consistent lists of functions will emerge providing at least a first approximation of what functions ought to be considered for particular types of wetlands. Also, the site selection is almost certainly constrained by logistics and finances making it difficult to ensure that the sites in the reference set are not somehow biased. As the approach is applied to novel sites within the region, preferably with some re-examination of indicator validity, it will become apparent whether the original set can in fact represent other sites in the same class of wetlands.

An unresolved issue in our example is how to deal with actual wetland size in assessing these functions. At this point we have estimated both size-independent and area-weighted scores for most functions although all data presented here are size-independent. The question simplifies to, "Is twice the area with half the function equal to half the area with twice the function?" Size-independent approaches do not inherently devalue small wetlands that may be particularly important in areas of higher human population. Avoiding simple area-weighting also recognizes the likely non-linear relationship between size of a wetland and potential value to a species or process. At the same time we realize that certain functions, flood attenuation for example, will be directly proportional to the size of the site and large sites, properly situated, are inherently more valuable than small sites for this function (see McAllister et al., 2000). It is possible that this decision will have to be made on a case-by-case basis and again a clear rationale will help avoid both confusion and confrontation.

In the present form of the HGM assessment and in our application the vast majority of effort is dedicated to the actual wetland site with relatively little collection of information about the surrounding landscape. It is likely that in many regions and for many functions the controlling factors will be larger-scale characteristics rather than site-specific. For a set of salt marshes in the northeast United States the proportion of residential land use was a good predictor of plant zonation and nitrogen processing (Wigand et al., 2001). These larger-scale variables may either prove to be better indicators than site characteristics or they may act as overall constraints on wetland functioning. Some wildlife habitat values are also related to surrounding uplands or even the proximity of other landscape elements.

Finally, there is always the risk of misapplication of models by applying relationships between indicators and functions to sites outside the domain where the relationship was established. Part of our intent in writing this overview was to highlight what we found to be the critical and difficult issues and questions in developing a reference data set. We hope others will add their experience, establishing a sort of minimum standard for useful application. While a considerable effort is required to develop locally-validated models and lists of functions, we believe the benefit of establishing a spatially extensive, broadly defined set of information on wetland functions justifies the effort.

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